available at www.sciencedirect.com
ScienceDirect
journal homepage: www.elsevier.com/locate/ecoleng

Effect of intermittent drainage on swine wastewater treatment by marsh–pond–marsh constructed wetlands

M.E. Poach[a,*], P.G. Hunt[a], G.B. Reddy[b], K.C. Stone[a], M.H. Johnson[a], A. Grubbs[b]

[a] Coastal Plains Soil, Water, and Plant Research Center, 2611 West Lucas Street, Florence, SC 29501, United States
[b] Department of Natural Resources and Environmental Design, North Carolina A&T State University, Greensboro, NC 27411, United States

ARTICLE INFO

Article history:
Received 30 December 2005
Received in revised form
14 November 2006
Accepted 25 January 2007

Keywords:
Constructed wetlands
Animal wastewater
Intermittent drainage
Marsh–pond–marsh design

ABSTRACT

The research objective was to investigate the effect of intermittent wetland drainage on swine wastewater treatment by marsh–pond–marsh (m–p–m) constructed wetlands. For 16 weeks beginning in June 2002, each of four m–p–m wetlands in Greensboro, NC, USA, received a different application of swine wastewater. The four application schemes were as follows: (1) continuous application; (2) 1 week of no application for every 4 weeks of application; (3) 1 week of no application for every 3 weeks of application; and (4) 1 week of no application for every 2 weeks of application. The effect of intermittent wetland drainage was determined by comparing each system's soil oxidation, wastewater constituent removal, and ammonia volatilization. Soil oxidation was increased during drainage periods of the systems with four and five drainage periods. While the removal of total suspended solids, chemical oxygen demand, and total phosphorus were not affected by the incorporation of drainage periods, the efficiency of total nitrogen removal significantly increased with increased number of drainage periods. For treatment wetlands that incorporated zero, three, four, and five drainage periods, the total nitrogen removal efficiencies were 57, 64, 70, and 67%, respectively. An increase in the number of drainage periods did not reduce ammonia volatilization from either marsh or pond sections.

© 2007 Elsevier B.V. All rights reserved.

1. Introduction

Even though constructed wetlands have been used for decades to treat municipal and industrial wastewater, the ability of this technology to treat animal wastewater has only recently been examined by significant research efforts (Hunt and Poach, 2001; Knight et al., 2000; Cronk, 1996). This research verified that constructed wetlands used in conjunction with land application were effective for treating animal wastewater, especially the removal of nitrogen and oxygen-demanding substances. Despite this effective treatment, the capacity of the constructed wetlands to remove wastewater nitrogen and oxygen-demanding substances was limited by the oxygen availability in the wastewater and wetland soil (Hunt et al., 2003).

Modifications of wetland operation can enhance the oxygen content of the wastewater and wetland soil; thereby, improving the removal of wastewater nitrogen and oxygen-demanding substances. Oxygen-enhancing modifications include intermittent wetland drainage (alternating fill and drain cycle). Intermittent drainage of wetland mesocosms treating dairy wastewater enhanced the removal of chemical

* *Corresponding author at.* Massachusetts Department of Environmental Protection, 436 Dwight Street, Springfield, MA 01103, United States. Tel.: +1 413 755 2128; fax: +1 413 784 1149.
E-mail address: matthew.poach@state.ma.us (M.E. Poach).
0925-8574/$ – see front matter © 2007 Elsevier B.V. All rights reserved.
doi:10.1016/j.ecoleng.2007.01.003

oxygen demand (COD) and nitrogen as compared to continuous application (Tanner et al., 1999). Although intermittent wetland drainage did not increase wetland removal of dairy wastewater phosphorus (Tanner et al., 1999), it did increase the removal of phosphorus in simulated wastewater (Busnardo et al., 1992). Therefore, phosphorus removal by constructed wetlands under intermittent drainage needs further study.

While oxygen input to wetland soil can be enhanced by the modification of wetland operation, oxygen input to the wastewater can be enhanced by the modification of wetland design. One of these design modifications, referred to as a marsh–pond–marsh (m–p–m) design, consists of a traditional continuous-marsh design bisected by a deeper, open-water or pond section. The pond section has been shown to increase the dissolved oxygen concentration and oxidation status of animal wastewater (Cathcart et al., 1994; Reddy et al., 2001). Although increased oxygen concentration was expected to promote nitrification in the pond section, m–p–m systems did not exhibit increased wastewater nitrogen removal when compared to continuous marsh systems (Poach et al., 2004b; Moore et al., 1995). This discrepancy suggests that the increased oxygen concentration did not support an increased rate of nitrification. The restriction on nitrification likely resulted from the application of wastewater biochemical oxygen demand (BOD) in excess of the first marsh's removal capacity (Poach et al., 2004b).

While the BOD treatment capacity of the first marsh in a m–p–m system can be increased by increasing the size of the first marsh, this modification may not be feasible for pre-existing systems. This modification is also not desirable for areas where animal wastewater treatment is land limited, especially when a goal of maximum nitrogen removal also maximizes BOD loading. One possible alternative for enhancing the treatment capacity of m–p–m systems is to apply wastewater on an alternating fill and drain cycle. If intermittent wetland drainage improves the BOD treatment capacity of the first marsh then the pond section should promote nitrification. Enhanced nitrification in the pond section should reduce ammonia (NH_3) volatilization generated by the pond (Poach et al., 2004a). Partial nitrification of swine wastewater before wetland application reduced NH_3 volatilization from continuous marsh wetlands (Poach et al., 2003).

The objective of this research was to evaluate how intermittent wetland drainage affects the ability of m–p–m constructed wetlands to treat wastewater from a confined swine operation. To meet this objective, different wastewater application schemes were used to load swine wastewater to four m–p–m wetland systems.

2. Materials and methods

2.1. Site description

The experiment was conducted at the swine research facility of the North Carolina A&T State University farm in Greensboro, NC, USA, using four surface-flow wetlands in a m–p–m design. The wetlands (11 m × 40 m) were constructed in 1995 (Reddy et al., 2001). Each wetland system (WS) consisted of

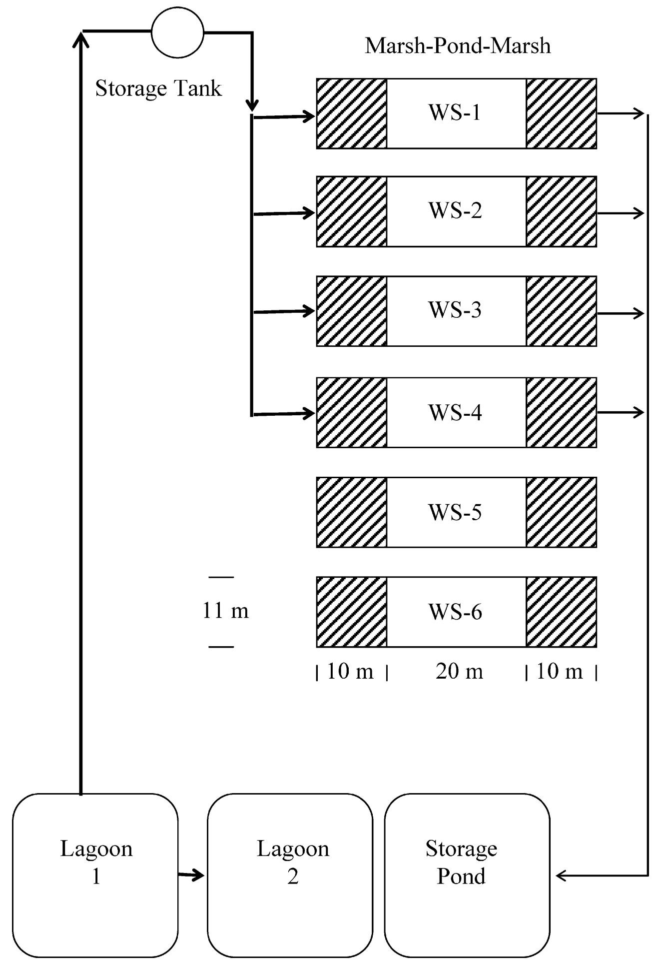

Fig. 1 – Schematic of the marsh–pond–marsh constructed wetland design showing the sources and flow paths for swine wastewater. Only wetland systems (WS) 1–4 were used in this experiment.

an 11 m × 10 m marsh at both the influent and effluent ends and an 11 m × 20 m pond section separating the marshes (Fig. 1). The marsh sections were planted with *Typha latifolia* L. (broadleaf cattail) and *Schoenoplectus americanus* (Pers.) Volkart ex Schinz & R. Keller (American bulrush) in March 1996. The wetlands have been treating swine wastewater every year since the first application in 1997.

2.2. Experimental design

The study was conducted for 16 weeks from June to October 2002. During the study, each WS received a different swine wastewater application scheme. The four application schemes were as follows: (1) continuous application (no drain periods); (2) 1 week of no application for every 4 weeks of application (three drain periods); (3) 1 week of no application for every 3 weeks of application (four drain periods); and (4) 1 week of no application for every 2 weeks of application (five drain periods). These application schemes resulted in drainage frequencies that were 0, 19, 25, and 31% of the total study period, respectively.

Wastewater applied to each WS was obtained from the primary lagoon (L_1) of a two-stage anaerobic lagoon that received manure flushed from the swine house (Fig. 1). Wastewater from L_1 was transferred by a submersible pump to an 8000 L storage tank, where it was discharged by gravity to all wetlands. Wastewater flows to each WS were controlled by ball valves. Effluent was discharged to an outlet basin located at the distal end of each WS before flowing to a storage pond. The wastewater inflows and outflows were measured with tipping buckets wired to an electronic totalizer (cycle counter). Periodically, mechanical flow measurements were verified manually.

While hydraulic loads to each system varied on a weekly basis due to the method of application, average loads for the study period were 8.5, 9.7, 9.0, and 8.3 $m^3 day^{-1}$ for wetlands with 0, 3, 4, and 5 drainage periods, respectively. Based on the volume of wastewater in each system, these loads should have resulted in retention times between 13 and 15 days.

Depth of water in each WS was controlled by the positioning of a vertical extension of the outflow pipe in the outlet basin. When a wetland was receiving wastewater, the operating heights of wastewater above the sediment surface were 15 cm in the marsh sections and 75 cm in the pond sections. During drain periods, the wastewater inflow was stopped and the pipe in the outlet basin was lowered to drop the water below the organic soil layer in the marsh sections. The maximal depth to which the wastewater could be drained was controlled by the depth of the drainage catchment basin in the second marsh of each system. The catchment basin, to which the outflow pipe was attached, was installed just below the mineral soil surface in each marsh to ensure the complete drainage of the organic layer.

The oxidation status of each marsh section was determined by measuring the soil oxidation–reduction potential (ORP). Soil ORP was measured using permanently installed electrodes. In 1997, after wetland construction, platinum (Pt) tipped electrodes and one reference electrode with a salt bridge assembly were inserted into the mineral sediment of each marsh section to depths of 5 and 15 cm. The Pt-tipped electrodes were constructed with 12-gauge copper (Cu) wire and 18-guage Pt wire. Construction was similar to the welded construction of Patrick et al. (1996) except that the Pt wire was not welded to the Cu wire but was instead inserted into a small hole drilled into the end of the Cu wire and held in place by crimping the end of the Cu wire around it. The reference electrode-salt bridge assembly was constructed according to Szogi et al. (2004). Prior to installation, electrodes were cleaned and tested according to Szogi et al. (2004).

A CR7X Campbell Scientific Datalogger (Logan, UT)[1] was used to record ORP readings in millivolts every 5 min and to average those readings on an hourly basis. The datalogger also recorded tipping bucket data and data collected by an onsite weather station.

[1] Mention of trade name, proprietary product, or vendor is for information only and does not constitute a guarantee or warranty of the product by U.S. Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that may also be suitable.

Discrete wastewater samples were collected from the inlet storage tank and from the outlet of each WS using autosamplers (ISCO 3700, Lincoln, NE). For each location, the samplers combined daily samples into weekly composites. Concentrated sulfuric acid was added to sampling bottles prior to sample collection to lower the pH below 2. At the end of each weekly sampling period, samples were transferred to the laboratory for analysis and stored at 4 °C.

During a 2-week period in September, NH_3 emitted from each marsh and pond section was sampled using a special open-ended enclosure. Plots for sampling NH_3 volatilization were located near the middle of each section. Poach et al. (2004a) give a detailed description of the enclosure design and the method used to sample NH_3. Two sampling runs were conducted on each plot except for the plots located in the second marsh sections of the first and fourth WS. For these plots, only one sampling run was conducted. Sampling runs were conducted for a 2-h period on marsh plots and for a 1-h period on pond plots.

2.3. *Data analyses*

Hourly ORP, tipping bucket, temperature, and rainfall records were downloaded from the datalogger and averaged on a 24-h basis. Daily ORP values were transformed to standard hydrogen electrode readings (Eh) by adding a +200-mv correction value associated with using a Ag/AgCl reference electrode. No further corrections to the Eh values were made as explained by Szogi et al. (2004). For each tipping bucket, daily values were multiplied by the volume of wastewater applied per tip cycle to obtain daily wastewater flow.

Weekly composite wastewater samples were analyzed for nitrate/nitrite-N (353.1), total Kjeldahl-N (351.2), and total P (365.4) using EPA methods (Kopp and McKee, 1983). These analyses were performed with a Technicon AAII analyzer (Technicon Instruments Corp., Tarrytown, NY). Total N was the sum of total Kjeldahl-N and nitrate/nitrite-N. Total suspended solids (TSS) of wastewater samples were determined as follows: a 20-mL aliquot of wastewater from each sample was filtered through a pre-dried glass fiber filter that was subsequently dried at 105 °C to constant weight. Chemical oxygen demand (COD) of wastewater samples was determined using the closed reflux, colorimetric method (5220; APHA, 1998).

For each wastewater constituent, total mass loaded to and discharged from each WS were calculated to determine mass removal. For these systems, mass removal is a more informative parameter than concentration reduction because regulations prevent the direct discharge of wastewater from confined animal operations. The mass of each wastewater constituent loaded to and discharged from each WS on a weekly basis was determined using the following equation:

$$M_L = \left(\frac{C_I}{10^6}\right) \times Q_I \tag{1}$$

$$M_D = \left(\frac{C_O}{10^6}\right) \times Q_O \tag{2}$$

where $M_{L/D}$ is the constituent mass loaded to or discharged from a WS (kg), $C_{I/O}$ the constituent concentration in waste lagoon or at WS outlet (mg L^{-1}) and $Q_{I/O}$ is the weekly total wastewater inflow or outflow (L).

For each wastewater constituent, total mass loaded to and discharged from each WS was calculated by summing the 16 weekly values obtained during the experimental period. Constituent mass removal efficiencies for each WS were determined using the following equation:

$$\mathrm{Eff} = \left[\frac{\mathrm{TM_L} - \mathrm{TM_D}}{\mathrm{TM_L}} \times 100\right] \quad (3)$$

where Eff is the mass removal efficiency (%), TM_L the total constituent mass loaded to a WS (kg), and TM_D is the total constituent mass discharged from a WS (kg).

Ammoniacal N in NH_3 volatilization samples was determined with a Bran + Luebbe Autoanalyzer III (Bran + Luebbe Analyzing Inc., Roselle, IL) using EPA method 351.2 (Kopp and McKee, 1983). Rates of NH_3 volatilization in mg NH_3-N m^{-2} h^{-1} were determined from the difference in NH_3-N collected by the inlet and outlet gas-washing bottles after adjusting for the air sampling ratio (Eq. (2); Poach et al., 2002).

2.4. *Statistical analysis*

The affect of drainage on mass removal of each wastewater constituent (TSS, COD, TP, and TN) was investigated with the regression procedure of the SAS system (SAS, 1990).

3. Results

3.1. *Water level*

For the WS with three drainage periods, the water level dropped 3.5 and 1.7 cm below the level of zero discharge during the first two drainage periods, respectively (Fig. 2a). The water level remained above the level of zero discharge during the final drainage period when a 29.4 mm rain event occurred. For the WS with four drainage periods, the water level dropped 4.9, 6.1, 1.9, and 1.6 cm below the level of zero discharge during the four drainage periods, respectively (Fig. 2b). During the first drainage period, a temporary increase in water level above the level of zero discharge occurred and coincided with a 9.9 mm rain event. For the WS with five drainage periods, the water level dropped 4.0, 2.7, 2.1, and 0.9 cm below the level of zero discharge during the first four drainage periods, respectively (Fig. 2c). Temporary increases in water level occurred during the second and third drainage periods and coincided with a 30.5 and a 19.6 mm rain event, respectively. The water level remained above the level of zero discharge during the final drainage period when a 29.4 mm rain event occurred.

3.2. *Soil oxidation*

For the WS with continuous wastewater application, Eh of the first marsh generally remained below zero (Fig. 3a). Increases in Eh occurred during rain events. While the Eh of the second marsh was around +100 mV at the start of the experiment, it slowly decreased to below zero during the next 4 weeks and

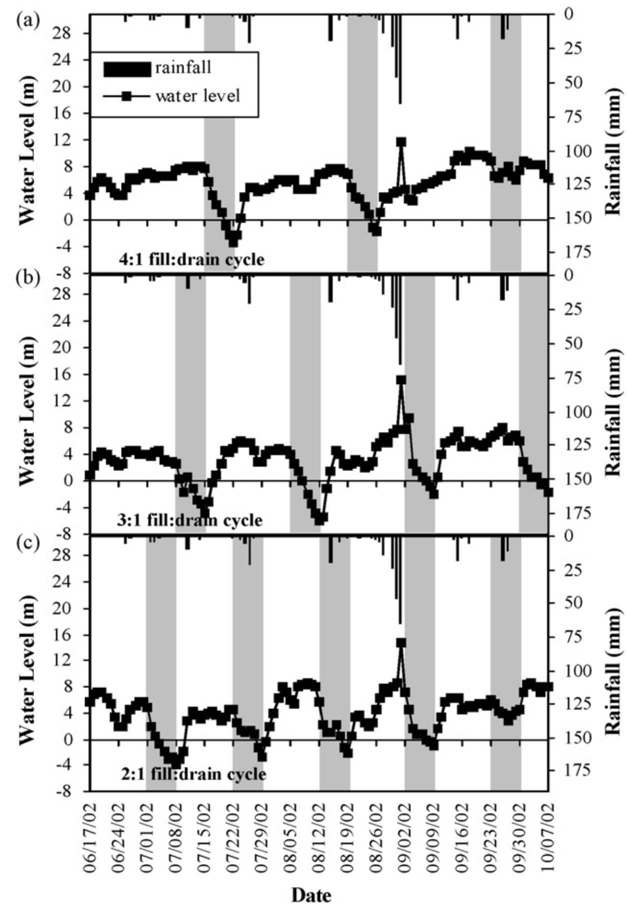

Fig. 2 – Wastewater levels in reference to the level of zero discharge for (a) the WS that had three drainage periods, (b) the WS that had four drainage periods, and (c) the WS that had five drainage periods. Grey bars indicate drainage periods.

tended to remain below zero. Increases in the Eh of the second marsh also occurred during rain events. The Eh's of both marshes remained below +190 mV.

For the WS with three drainage periods, the oxidation status the first marsh could not be assessed because of faulty ORP electrodes. The Eh of the second marsh started below zero and tended below zero throughout the experiment (Fig. 3b). Increases in Eh of this marsh occurred during rain events with two such events coinciding with the last two drainage periods. The Eh never increased above +16 mV.

For the WS with four drainage periods, the Eh of the first marsh increased from +30 to +302 mV during the first drainage period and from −50 to +319 mV during the second drainage period (Fig. 3c). The Eh did not increase during the last two drainage periods. The Eh of the second marsh was approximately 100 mV at the start of the experiment and fluctuated between −40 and +165 mV throughout the experiment. Positive spikes in Eh occurred during rain events. An increase in Eh from −5 to +130 mV occurred during the first drainage period and coincided with a rain event. During the second drainage period, Eh increased only slightly from +1 to +52 mV. The Eh decreased from +103 to +17 then increased to +63 mV during

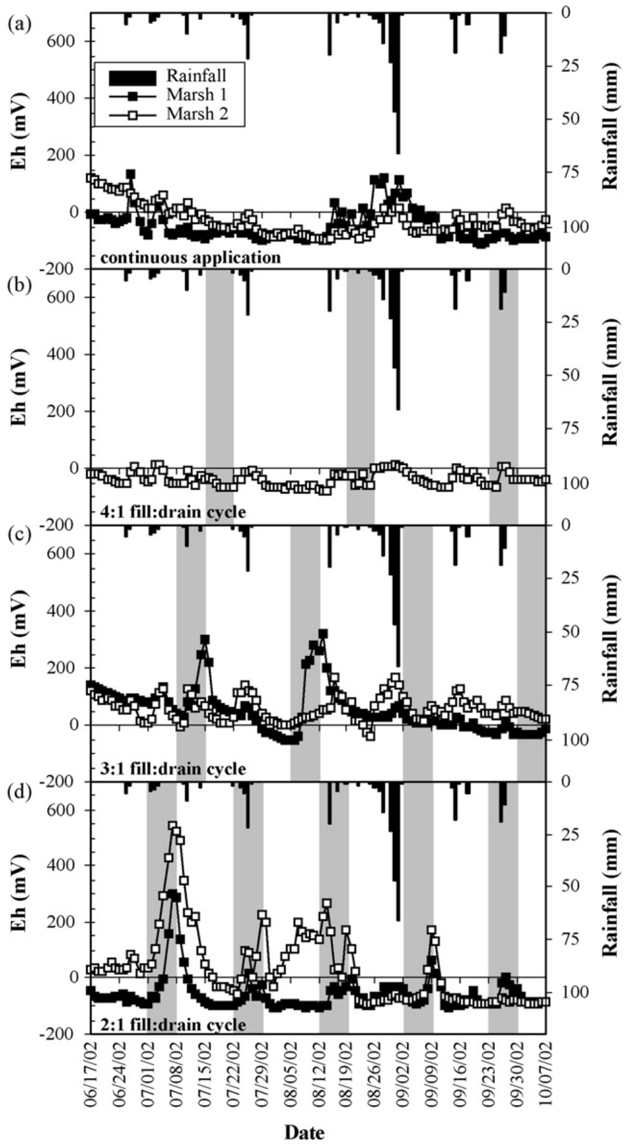

Fig. 3 – Soil electrical potential (standard hydrogen reference) for the first and second marshes of (a) the WS that had continuous wastewater application, (b) the WS that had three drainage periods, (c) the WS that had four drainage periods, and (d) the WS that had five drainage periods. Grey bars indicate drainage periods.

the third drainage period. The largest rain event during the experiment (65.5 mm) occurred a day before the third drainage period. No increase in Eh occurred during the last drainage period.

For the WS with five drainage periods, the Eh of both marsh sections increased during all but the last drainage period when Eh increased in the first marsh only (Fig. 3d). Both marsh sections exhibited the greatest increase in Eh during the first drainage period reaching +304 in the first marsh and +544 mV in the second marsh. From lows of around −90 mV, the Eh of the first marsh increased to maximums of +15, −6, +62, and +1 mV during drainage periods 2–5, respectively. The Eh of the second marsh increased to maximums of +228, +268, and +169 mV during drainage periods 2–4, respectively. The Eh of both marshes peaked twice during each of the second and third drainage periods. The decrease in Eh during those drainage periods coincided with rain events. The largest rain event during the experiment (65.5 mm) occurred a day before the fourth drainage period. A rain event also occurred during the final drainage period.

3.3. *TSS and COD*

During the study, the average concentration of TSS applied to the wetlands was 160 mg L^{-1} (Table 1), which resulted in 2899, 3111, 2674, and 2457 kg ha^{-1} of TSS being applied to the wetlands with zero, three, four, and five drainage periods, respectively (Table 2). The wetlands with zero, three, four, and five drainage periods had average outlet TSS concentrations of 81, 118, 105, and 105 mg L^{-1}, respectively.

The average concentration of COD applied to the wetlands was 445 mg L^{-1} (Table 1), which resulted in 8002, 8763, 7393, and 6604 kg ha^{-1} of TSS being applied to the wetlands with zero, three, four, and five drainage periods, respectively (Table 2). The wetlands with zero, three, four, and five drainage periods had average outlet COD concentrations of 246, 321, 285, and 287 mg L^{-1}, respectively.

When incorporating zero, three, four, and five drainage periods during the experimental period, the wetland systems removed 62, 51, 63, and 62% of mass TSS and 59, 51, 61, and 56% of mass COD, respectively (Table 2). Regression analysis indicated that intermittent wetland drainage did not have a significant effect on wetland removal of either TSS ($p = 0.977$) or COD ($p = 0.859$).

3.4. *Phosphorus and nitrogen*

During the study, the average concentration of total P applied to the wetlands was 71 mg L^{-1} (Table 1), which resulted in 1231, 1378, 1207, and 1049 kg ha^{-1} of total P being applied to the wetlands with zero, three, four, and five drainage periods, respectively (Table 2). The wetlands with zero, three, four, and five drainage periods had average outlet total P concentrations of 66, 68, 62, and 68 mg L^{-1}, respectively.

The average concentration of total N applied to the wetlands was 66 mg L^{-1} (Table 1), which resulted in 1374, 1424, 1198, and 1086 kg ha^{-1} of total N being applied to the wetlands with zero, three, four, and five drainage periods, respectively (Table 2). The wetlands with zero, three, four, and five drainage periods had average outlet total N concentrations of 30, 32, 27, and 30 mg L^{-1}, respectively.

When incorporating zero, three, four, and five drainage periods during the experimental period, the wetland systems removed 29, 33, 46, and 34% of total P mass and 57, 64, 70, and 67% of total N mass, respectively (Table 2). While regression analysis did not indicate a significant effect of intermittent wetland drainage on the wetland removal of total P mass ($p = 0.442$), it did show a significant trend of increased efficiency of total N mass removal with an increased number of drainage periods ($p = 0.098$).

Rates of NH_3 volatilization ranged from 0.7 to 20 mg N m^{-2} h^{-1} for marsh sections and ranged from 10 to 92 mg N m^{-2} h^{-1} for pond sections (Fig. 4). For marsh

Table 1 – Mean (±1S.D.) concentrations for total suspended solids (TSS), chemical oxygen demand (COD), total phosphorus (P), and total nitrogen (N) in the influent to and effluent from marsh–pond–marsh treatment wetlands to which swine wastewater was applied continuously and with incorporated drain periods[a]

Wastewater constituent	Influent concentration ($mg\,L^{-1}$)	Effluent concentration ($mg\,L^{-1}$)			
		Continuous	Three drain periods	Four drain periods	Five drain periods
TSS	160 ± 28	81 ± 16	118 ± 77	105 ± 33	105 ± 28
COD	445 ± 70	246 ± 35	321 ± 118	285 ± 43	287 ± 52
Total P	71 ± 10	66 ± 8	68 ± 6	62 ± 11	68 ± 12
Total N	66 ± 13	30 ± 12	32 ± 11	27 ± 6	30 ± 16

[a] One-week drain periods incorporated on a consistent basis over a 16-week experimental period.

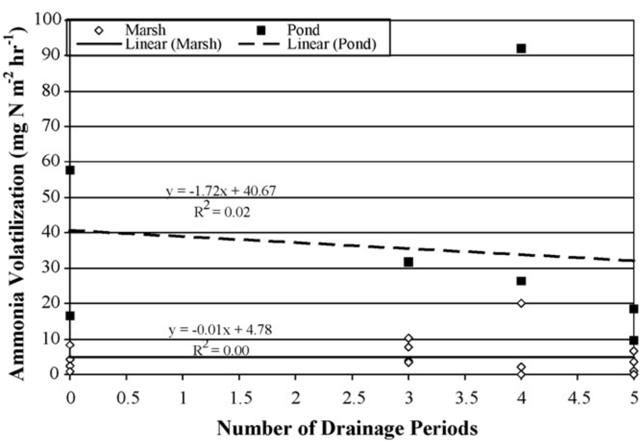

Fig. 4 – Regression of ammonia volatilization vs. number of drainage periods for marsh and pond sections of wetlands that received four different wastewater application schemes.

and pond sections, regression analysis indicated that NH_3 volatilization was not effected by intermittent wetland drainage ($p = 0.72$ and $p = 0.72$ for marsh and pond sections, respectively).

4. Discussion

While intermittent wetland drainage was expected to promote aerobic soil conditions (Eh ≥ +200 mV) during the drainage periods, these conditions were not detected during every drainage period. For the WS that had five drainage periods aerobic soil conditions were only detected during three drainage periods, with soil Eh of both marshes reaching a maximum during the first drainage period. Despite soil Eh increasing during each drainage period, the Eh levels exhibited during the first drainage period were not replicated during subsequent drainage periods as a result of incomplete soil drainage. Complete soil drainage was prevented due to rain events that occurred either immediately prior to or during the final four drainage periods.

Aerobic soil conditions were also not detected during all drainage periods of the WS that had four drainage periods. Aerobic soil conditions were only detected during the first two drainage periods and then only in the first marsh. Aerobic soil conditions were not detected in the first marsh during the last two drainage periods because the water level did not drop as far as it dropped during the first two drainage periods (Fig. 2b). The depth of water drainage could also explain why aerobic soil conditions were not detected in the second marsh if the ORP electrodes were located below that depth. It is also possible that the sediment of the second marsh retained enough moisture after drainage to inhibit aerobic conditions. The fact that aerobic soil conditions were not detected in the second marsh of the WS that had three drainage periods could also be explained by remnant soil saturation or the water level not dropping below the depth of the ORP electrodes during drainage periods.

Due to ORP electrode placement, the inability to detect aerobic soil conditions during drainage periods does not preclude the existence of aerobic soil conditions during those periods. The ORP electrodes were originally placed in the mineral soil of the wetlands around the time of wetland construction. Since that time, the mineral sediments had been overlain by an 8- to 10-cm layer of organic material. It is possible that the wastewater level only dropped to a depth where the organic layer was drained while the mineral layer was not.

Despite the production of aerobic soil conditions by intermittent wetland drainage, an increase in the number of drainage periods did not effect wetland treatment of TSS. The lack of an effect on TSS treatment was not unexpected because wetlands remove wastewater TSS by mainly physical processes independent of soil oxidation status.

An increase in the number of drainage periods also showed no effect on wetland treatment of COD, even though it was expected to improve COD removal. In a previous study of treatment wetlands, intermittent wetland drainage improved treatment wetland removal of COD (Tanner et al., 1999). While treatment wetlands remove wastewater COD through a combination of biochemical and physical conversions of organic compounds and ammonia (Kadlec and Knight, 1996), COD can also increase in treatment wetlands that contain open water through the growth of floating plants and algae (Poach et al., 2004b; Cathcart et al., 1994). In the present study, it is possible that any reduction in COD as a result of intermittent wetland drainage was offset by increased COD production in the wetlands' pond sections.

An increase in the number of drainage periods had no effect on wetland treatment of total P, even though intermittent wetland drainage did improve the removal of P in a previous treatment wetland study (Busnardo et al., 1992). In that previous study, the increased removal of P was supposed to occur as a result of increased formation of iron oxyhydroxides in

Table 2 – Efficiencies for mass removal of total suspended solids (TSS), chemical oxygen demand (COD), total phosphorus (P), and total nitrogen (N) by marsh–pond–marsh treatment wetlands to which swine wastewater was applied continuously and with incorporated drain periods[a]

Wastewater constituent	Continuous application			Three drain periods			Four drain periods			Five drain periods		
	Mass in ($kg\,ha^{-1}$)	Mass out ($kg\,ha^{-1}$)	Removal (%)	Mass in ($kg\,ha^{-1}$)	Mass out ($kg\,ha^{-1}$)	Removal (%)	Mass in ($kg\,ha^{-1}$)	Mass out ($kg\,ha^{-1}$)	Removal (%)	Mass in ($kg\,ha^{-1}$)	Mass out ($kg\,ha^{-1}$)	Removal (%)
TSS	2899	1115	62	3111	1519	51	2674	998	63	2457	944	62
COD	8002	3280	59	8763	4323	51	7393	2857	61	6604	2916	56
Total P	1231	871	29	1378	923	33	1207	647	46	1049	691	34
Total N	1374	597	57	1424	506	64	1198	357	70	1086	363	67

[a] One-week drain periods incorporated on a consistent basis over a 16-week experimental period.

the mineral sediment. Because the treatment wetlands were surface flow, most of the wastewater treatment occurred in the organic soil layer, which would have precluded increased P removal by increased iron oxyhydroxide formation.

An increase in the number of drainage periods improved m–p–m wetland removal of total N. This improvement was expected because increased soil oxidation would have increased the nitrification of wastewater ammonia, which is the process that limits the removal of total N in m–p–m wetlands that treat swine wastewater (Hunt et al., 2003). While nitrification limits the removal of total N, denitrification is also necessary for N removal and requires anaerobic soil conditions. Because anaerobic conditions are also required for N treatment, N removal efficiency cannot be expected to continue to increase with drainage frequencies greater than investigated in this study.

Because both aerobic and anaerobic soil conditions are necessary for N removal from wastewater, intermittent wetland drainage was performed in this study to create temporal variations in soil oxidation status. Soil oxidation status can also be varied spatially by discontinuous wastewater application (intermittent cessation of wastewater application). Spatial variation in soil oxidation status requires a wetland with a sloped grade. On a wetland with a sloped grade, discontinuous wastewater application can be used to create aerated conditions in the upper section of the wetland while the lower section remains flooded and thereby reduced. With such a design, aerobic soil conditions may not be impacted as greatly by rain events as systems with a level grade.

Despite an increase in nitrification as indicated by the increased total N treatment, an increase in the number of drainage periods had no effect on NH_3 volatilization. The reduction of NH_3 volatilization from the pond sections of m–p–m wetlands treating swine wastewater required an improvement of BOD removal by the first marsh. The lack of an effect of drainage on COD treatment indicated that drainage also did not improve BOD removal. Because intermittent wetland drainage did not alleviate NH_3 volatilization from the pond sections and because drainage can likely be better controlled on sloped systems, continuous marsh wetlands should be preferred over m–p–m wetlands when using intermittent wetland drainage to treatment wastewater.

5. Conclusions

Results indicated that intermittent wetland drainage in the summer and early fall has the potential to promote aerobic soil conditions during periods when the wetlands are completely drained. Incomplete soil drainage inhibited aerobic soil conditions at the mineral soil level, but such conditions could have been produced in the organic soil layer above that level. Incomplete soil drainage resulted from rain events that occurred during drainage periods.

Despite the production of aerobic soil conditions by intermittent wetland drainage, an increase in the number of drainage periods had no effect on wetland removal of TSS, COD, and total P. Wetland removal of total N tended to increase with increased number of drainage periods. This increase in total N removal likely resulted from increased nitrification of wastewater ammonia. Despite the increased total N treatment, an increase in the number of drainage periods had no effect on NH_3 volatilization from either the marsh or pond sections of the wetlands.

REFERENCES

APHA, 1998. Standard Methods for the Examination of Water and Wastewater, 20th ed. American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, DC.

Busnardo, M.J., Gersberg, R.M., Langis, R., Sinicrope, T.L., Zedler, J.B., 1992. Nitrogen and phosphorus removal by wetland mesocosms subjected to different hydroperiods. Ecol. Eng. 1, 287–307.

Cathcart, T.P., Hammer, D.A., Triyono, S., 1994. Performance of a constructed wetland—vegetated strip system used for swine waste treatment. In: DuBowy, P.J., Reaves, R.P. (Eds.), Constructed Wetlands for Animal Waste Management. Purdue Research Foundation, West Lafayette, IN, USA, pp. 9–22.

Cronk, J.K., 1996. Constructed wetlands to treat wastewater from dairy and swine operations: a review. Agric. Ecosyst. Environ. 58, 97–114.

Hunt, P.G., Poach, M.E., 2001. State of the art for animal wastewater treatment in constructed wetlands. Water Sci. Technol. 44 (11–12), 19–25.

Hunt, P.G., Matheny, T.A., Szogi, A.A., 2003. Denitrification in constructed wetlands used for the treatment of swine wastewater. J. Environ. Qual. 32, 727–735.

Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. Lewis Publishers, Boca Raton, FL, USA, p. 893.

Knight, R.L., Payne Jr., V.W.E., Borer, R.E., Clarke Jr., R.A., Pries, J.H., 2000. Constructed wetlands for livestock wastewater management. Ecol. Eng. 15, 41–55.

Kopp, J.F., McKee, G.D., 1983. Methods for chemical analysis of water and wastes. USEPA Report No. EPA-600/4-79020. Environmental Monitoring and Support Lab, Office of Research and Development, U.S. EPA, Cincinnati, OH, USA, 521 pp.

Moore, J.A., Gamroth, M.J., Skarda, S.M., Niswander, S.F., 1995. Treating dairy flush water in a constructed wetland. In: Proceedings of the Seventh International Symposium on Agricultural and Food Processing Wastes, June 18–20, American Society of Agricultural Engineers, Chicago, IL, USA, pp. 74–86.

Patrick, W.H., Gambrell, R.P., Faulkner, S.P., 1996. Redox measurements of soils. In: Sparks, D.L. (Ed.), Methods of Soil Analysis: Chemical Methods, Part 3. Soil Science Society of America, Madison, WI, pp. 1255–1273.

Poach, M.E., Hunt, P.G., Sadler, E.J., Matheny, T.A., Johnson, M.H., Stone, K.C., Huminek, F.J., Rice, J.M., 2002. Ammonia volatilization from constructed wetlands that treat swine wastewater. Trans. ASAE 45, 619–627.

Poach, M.E., Hunt, P.G., Vanotti, M.B., Stone, K.C., Matheny, T.A., Johnson, M.H., Sadler, E.J., 2003. Improved nitrogen treatment by constructed wetlands receiving partially nitrified liquid swine manure. Ecol. Eng. 20, 183–197.

Poach, M.E., Hunt, P.G., Reddy, G.B., Stone, K.C., Matheny, T.A., Johnson, M.H., Sadler, E.J., 2004a. Ammonia volatilization from marsh–pond–marsh constructed wetlands treating swine wastewater. J. Environ. Qual. 33, 844–851.

Poach, M.E., Hunt, P.G., Reddy, G.B., Stone, K.C., Johnson, M.H., Grubbs, A., 2004b. Swine wastewater treatment by marsh–pond–marsh constructed wetlands under variable nitrogen loads. Ecol. Eng. 23, 165–175.

Reddy, G.B., Hunt, P.G., Phillips, R., Stone, K., Grubbs, A., 2001. Treatment of swine wastewater in marsh–pond–marsh constructed wetlands. Water Sci. Technol. 44 (11–12), 545–550.

SAS, 1990. SAS Version 6.07. SAS Institute, Cary, NC, USA.

Szogi, A.A., Hunt, P.G., Sadler, E.J., Evans, D.E., 2004. Characterization of oxidation–reduction processes in constructed wetlands for swine wastewater treatment. Appl. Eng. Agric. 20 (2), 189–200.

Tanner, C.C., D'Eugenio, J., McBride, G.B., Sukias, J.P.S., Thompson, K., 1999. Effect of water level fluctuation on nitrogen removal from constructed wetland mesocosms. Ecol. Eng. 12, 67–92.